Two Essays on the Trade-Offs Between Multiple Policy Objectives of Environmental Management Efforts

by

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ABSTRACT

Environmental agencies often want to accomplish additional objectives beyond their central environmental protection objective. This is laudable; however it begets a need for understanding the additional challenges and trade-offs involved in doing so. The goal of this thesis is to examine the trade-offs involved in two such cases that have received considerable attention recently. The two cases I examine are (1) the protection of multiple environmental goods (e.g., bundles of ecosystem services); and (2) the use of payments for ecosystem services as a poverty reduction mechanism. In the first case (chapter 2), I build a model based on the fact that efforts to protect one environmental good often increase or decrease the levels of other environmental goods, what I refer to as "cobenefits" and "disbenefits" respectively. There is often a desire to increase the cobenefits of environmental protection efforts in order to synergize across conservation efforts; and there is also a desire to decrease disbenefits because they are seen as negative externalities of protection efforts. I show that as a result of reciprocal externalities between environmental protection efforts, environmental agencies likely have a disincentive to create cobenefits, but may actually have an incentive to decrease disbenefits. In the second case (chapter 3), I model an environmental agency that wants to increase environmental protection, but would also like to reduce poverty. The model indicates that in theory, the trade-offs between these two goals may depend on relevant parameters of the system, particularly the ratio of the price of monitoring to participant's compliance cost. I show that when the ratio of monitoring costs to compliance cost is higher, trade-offs between environmental protection and poverty reduction are likely to be smaller. And when the ratio of monitoring costs to compliance costs is lower, trade-offs are likely to be larger. This thesis contributes to a deeper understanding of the trade-offs faced by environmental agencies that want to pursue secondary objectives of protecting additional environmental goods or reducing poverty.

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Chapter 1

BACKGROUND

The Millennium Ecosystem Assessment created a strong guide for research related to the environment and human well-being. It established a number of key points: (1) Human well-being is closely tied to the benefits provided by ecosystems (ecosystem services); (2) many of the services provided by ecosystems are in decline; (3) there is a need for increased policy and management interventions to reverse the degradation of ecosystems and improve human well-being; and (4) in most cases we do not have sufficient understanding of ecosystem properties and how they contribute to human well-being to create effective interventions for reversing degradation and enhancing ecosystem services (*Millennium Ecosystem Assessment*, 2005).

This has spurred interest in addressing these gaps in scientific knowledge and increased the demand for effective policy to manage ecosystems (Carpenter *et al.*, 2009). In an effort to set the research agenda, considerable work has been done to clarify the nature of these gaps in our knowledge about ecosystems and their links with human well-being. In general, there has been a call to generate a richer understanding of the complex social-ecological context in which interactions between humans and ecosystems take place. For example, Carpenter *et al*. (2009) state that "New research is needed that considers the full ensemble of processes and feedbacks, for a range of biophysical and social systems, to better understand and manage the dynamics of the relationship between humans and the ecosystems on which they rely." Daily and Matson (2008) come to a similar conclusion. They state that advances in the management of ecosystems are needed in three key areas: (1) the science of ecosystem service production functions and the mapping of services; (2) the design of appropriate finance, policy and governing systems; and (3) the art of implementing these [programs] in diverse biophysical and social contexts.

In broad terms, the goal of this thesis is to contribute to the effort to understand challenges created by the context in which environmental protection efforts take place. In particular it focuses on situations where the biophysical and social context of environmental management efforts results in situations where there are likely to be multiple policy objectives and a need for understanding the trade-offs between these objectives. This thesis takes a partial equilibrium approach to examining these trade-offs. More complex impacts on markets and other factors that would be captured by a general equilibrium approach are not considered here. Specifically, the thesis focuses on two themes where this is the case: the management of multiple environmental goods (chapter 1); and efforts to use payments for ecosystem services (PES) as poverty alleviation mechanisms.

The first chapter in this thesis examines challenges faced by efforts to protect multiple environmental goods simultaneously. Because the benefits humans receive from ecosystems are often closely interconnected and share drivers such as land-use change (*Millennium Ecosystem Assessment*, 2005; Bennett *et al*., 2009) efforts to protect one environmental good can increase or decrease other environmental goods (what I refer to as cobenefits and disbenefits, respectively). This has generated a strong interest in deepening our understanding of when environmental protection efforts can achieve multiple environmental goals simultaneously and what trade-offs may exist between them. For example, the United Nation's Reducing Emissions from Deforestation and Forest Degradation (REDD) program has generated considerable interest in this regard. Venter *et al*. (2009) and Miles and Kapos (2008) both call for increasing the extent to which the massive transfer of money toward tropical nations increases biodiversity protection and other ecosystem services, not just carbon sequestration.

Considerable work has been done to identify, map and value multiple ecosystem services. For example, there is a variety of papers that seek to guide conservation by illustrating where on the landscape bundles of ecosystem services are produced, and thus where it might be best to focus conservation investments (e.g., Chan *et al*., 2006; Egoh *et al*., 2008; Naidoo *et al*., 2008). Spatially explicit modeling tools for conservation planning, such as InVEST (Nelson *et al*., 2009), have also been developed. InVEST uses ecological production functions and economic valuation techniques to assess the outcomes for multiple ecosystem services under different land-use/land-cover projections. Tools such as InVEST represent a significant step forward in coordinating conservation planning so that it takes into account the complex biophysical and social elements pertinent to effective environmental management.

While there has been considerable progress in conservation planning related to cobenefits and disbenefits, there is still a need for greater understanding of the challenges inherent in pursuing multiple policy objectives in this context. To contribute to this understanding, the first chapter in this thesis models a situation where there are two environmental agencies, each seeking to reach a fixed environmental goal for a different environmental good. In the model, protection efforts of each agency impacts both environmental goods. This allows for an examination of the incentives that each agency has to increase the degree to which its protection efforts benefit or diminish the second environmental good. The model shows that this situation results in reciprocal externalities between the two environmental agencies, which may create challenges for efforts to increase cobenefits and decrease disbenefits of environmental protection efforts. In this sense, the model highlights social feedbacks that may results from the context in which environmental protection efforts take place.

The second chapter of this thesis focuses on efforts to achieve environmental protection in locations where there is also concern about reaching development goals. The chapter considers a case where there is an environmental agency that would like to increase environmental protection, but would also like to achieve a reduction in poverty levels where the program is located. Naturally the goals of environmental protection and poverty reduction are both important goals; however there is a need to understand when these goals may overlap and what trade-offs exist between them.

The hope of achieving environmental protection and poverty reduction simultaneously is not new. Indeed, there is a long history of interconnections between environmental protection efforts and rural development initiatives.

Community-based natural resource management (CBNRM) is one example of this. The latter part of the twentieth century saw a sharp rise in efforts by national governments and conservation and development agencies to strengthen local institutional capacity in an effort to improve environmental management and stimulate rural development simultaneously. One common example of a CBNRM program is Zimbabwe's Communal Areas Management Programme for Indigenous Resources (CAMPFIRE), which was designed in the mid-1980s to stimulate rural development by devolving to communities the rights to manage and benefit from natural resource such as large game and their habitat (Frost & Bond, 2008; Taylor, 2009). CBNRM programs have met with mixed success. In some cases, such as that of CAMPFIRE, there have been identifiable improvements in wildlife management and economic gains for the community, however in many cases programs have had little identifiable success or there is too little information about revenue and community costs to determine the net effect of the program (Gibson & Marks, 1995).

More recently market-like payments for ecosystem services (PES) programs have become the mechanism of choice for environmental protection efforts in the developing world (Wunder, 2007). Not surprisingly, there is also hope that PES programs will function as rural development mechanisms. In general, PES programs work by offering landowners financial or in-kind compensation for managing lands in a way that ensures the provision of some environmental good. More specifically, Wunder (2005) defines a properly functioning PES as, "a voluntary transaction where a well-defined ES [ecosystem service] (or a land-use likely to secure that service) is being ‘bought’ by a (minimum one) ES buyer from a (minimum one) ES provider if and only if the provider secures ES provision (conditionality).” Because PES programs provide an income source to the landowners, there has been hope that PES would achieve environmental conservation and rural development simultaneously (Engel *et al*., 2008). However, as with the case of CBNRM, the value of PES as a poverty alleviation mechanism is highly context dependent, and thus far only limited success has been documented (Pagiola *et al*., 2005; Engel *et al*., 2008; Wunder, 2008).

Given the history of conservation and development efforts, the need to understand the trade-offs between environmental protection and poverty alleviation has been recognized for some time. However for a variety of reasons, there is an increasing urgency for understanding these trade-offs in the context of PES. One reason is that the perceived need for environmental protection in developing countries is increasing, particularly as climate change becomes a larger focus of environmental efforts. A significant portion of global carbon emissions come from deforestation in developing countries (DeFries *et al*., 2002). This has spurred an increased effort to reduce threats to carbon stocks in poorer nations, such as the UN’s proposed Reducing Emissions from Deforestation and Forest Degradation (REDD) program. REDD, which would provide financial incentives to countries to reduce their deforestation rates, would involve an unprecedented transfer of money to developing countries (Venter *et al*., 2009), and much of these funds would likely be channeled into PES-like mechanisms

(Pattanayak *et al.*, 2010). In addition, because the benefit of carbon emission reductions is essentially independent of the location of the emission reductions, carbon credit markets can target wherever emission abatement can be achieved at lowest cost, which may incentivize industries from developed countries to purchase offsets derived from reduced deforestation in developing countries.

The increased flow of conservation dollars to poor countries and the growing interest in achieving environmental and poverty goals simultaneously means there is a strong need for understanding the trade-offs between these goals. The possibility that "pro-poor premia" on environmental efforts that also achieve poverty reduction may increase the amount of funding available for environmental protection (Greig-Gran *et al.*, 2005) provides even further motivation. To help address this need, the second chapter in this thesis generates some theoretical insights by modeling a PES program where the environmental agency seeks to achieve both an environmental protection goal and a poverty reduction goal. This allows for an analysis of when these two goals may overlap and when there may be trade-offs.

In summary, the biophysical and social context of environmental management efforts often spurs the desire to address multiple policy objectives at the same time. The primary goal of this thesis is to provide some theoretical insights about the challenges of achieving multiple policy objectives in the cases discussed above, especially the trade-offs that may exist between these objectives.

Chapter 2

MANAGING FOR MULTIPLE ENVIRONMENTAL GOODS: THE CASE OF COBENEFITS AND DISBENEFITS OF ENVIRONMENTAL PROTECTION EFFORTS

Summary

Environmental agencies' efforts to protect one environmental good often increase or decrease the levels of other environmental goods as well. I refer to these as "cobenefits" and "disbenefits" of environmental protection efforts. There is interest in increasing the cobenefits of environmental protection efforts because they are seen as a way to synergize across conservation efforts; and there is interest in decreasing disbenefits because they are seen as negative externalities of protection efforts that could potentially be minimized. Because increasing cobenefits and decreasing disbenefits may be a way to increase the efficiency of environmental protection efforts, it is important to understand what incentives environmental agencies have to increase cobenefits and decrease disbenefits. This chapter examines this issue by building a model of two environmental goods and two environmental agencies (e.g., environmental NGOs or national governments), where each agency is focused on protecting one of the goods. The model is used to illustrate how increasing cobenefits and decreasing disbenefits impact the cost to the agencies of meeting a fixed environmental target. The model shows that the presence of cobenefits and disbenefits results in reciprocal externalities between the two agencies' environmental protection efforts. In the case of cobenefits, it is

shown that the agencies have a disincentive to increase cobenefits, whereas in the case of disbenefits the agencies have an incentive to decrease disbenefits. This work provides some basic theoretical insights pertinent to the management of multiple environmental goods when there are cobenefits and disbenefits of environmental protection efforts.

Introduction

It is well understood that efforts to protect one environmental good can increase or decrease other environmental goods. For example, efforts to sequester carbon by protecting natural forests can also protect biodiversity (and vice versa—efforts to protect biodiversity may also sequester carbon). These are often called "synergies" between protection efforts, but I will instead use the term "cobenefits" of protection efforts because this term is more clear and because this is term often used in the international conservation arena (Miles & Kapos, 2008). On the other hand, efforts to protect one environmental good may instead decrease other environmental goods—often called "trade-offs" between protection efforts. I will refer to these as "disbenefits" of conservation efforts. For example, Jackson *et al*. (2005) showed that afforestation (a strategy frequently used to sequester carbon) can result in substantial losses in stream flow and increased soil salinization. There is a call to increase cobenefits and decrease disbenefits; cobenefits are seen as a way to synergize across conservation efforts, and disbenefits are seen as negative externalities that should be minimized where possible. For example Venter *et al*. (2009) and Miles & Kapos (2008) both claim that the UN's REDD program should prioritize carbon sequestration opportunities that increase cobenefits like the protection of biodiversity and other ecosystem services.

It seems that there is a need to clarify some of the terminology used when discussing so-called "synergies" and "trade-offs" between environmental goods/ecosystem services. I think that the terms cobenefits and disbenefits are preferred terms for discussing the marginal impacts of conservation efforts. For example, supposed the budget dedicated to conservation is increased. If another dollar is spent to convert a patch of land from a non-conservation land-use to a conservation land-use, these terms help specify how much of two desired outcomes (say carbon sequestration and biodiversity conservation) is achieved. It may be that the net impact of the land-use change increases a targeted environmental good and also increases another environmental good (the case of cobenefits). Or it may be that the net impact of the land-use change is positive for the targeted good but negative for the other good (the case of disbenefits). By using these terms, one can differentiate cobenefits and disbenefits from the more traditional notion of trade-offs in economics where one is determining how much of two desired goods can be produced given scare resources/budget. This corresponds to a situation where there is instead a fixed conservation budget and one is determining how much of two desired environmental outcomes can be achieved given different allocations of the budget. Thus it is possible that for two environmental goods generally associated with cobenefits (say, carbon and biodiversity), there are still trade-offs between the two goods. It seems to me that this distinction is frequently glossed over in the literature, which obfuscates the discussion about protecting multiple ecosystem services/environmental goods.

Many papers have focused on determining the spatial pattern of cobenefits and disbenefits. Frequently these analyses identify areas where there is high spatial correlation between ecosystem services or areas that are priorities because of their contribution to multiple services. These studies have revealed that in some cases ecosystem services are spatially correlated, but that in other cases they may not be. For example, Anderson *et al*. (2009) map the correlation among areas important for biodiversity (richness of species of conservation concern) and other ecosystem services in Britain. They found that in some cases, those areas that are best for conserving species of concern also overlap with areas important for producing other ecosystem services, but in other cases they do not, and that this can vary considerably depending on the region of analysis. Chan *et al*. (2006) find that in central coastal California, biodiversity shows a weak positive correlation with some services (e.g., water provisioning, carbon storage) and a weak negative correlation with other services (pollination and forage production), but that strategic selection of associated services shows promise for meeting multiple ecosystem service protection goals. In a study focused on identifying bundles of ecosystem services in Quebec, Canada Raudsepp-Hearne *et al*. (2010) identify bundles of ecosystem services by mapping the spatial pattern of 12 ecosystem services and make suggestions for maximizing the protection of multiple services. They found that there are often trade-offs between provisioning and regulating services, but that greater diversity of ecosystem services is positively correlated with regulating ecosystem services.

Increasing cobenefits and decreasing disbenefits may be a way to increase the efficiency of conservation efforts (Nelson *et al*., 2008) and there is increasing information available to environmental agencies interested in doing so. For example it may be possible for an environmental agency to sequester carbon via a number of different strategies, each with different levels of cobenefits. Afforestation using monoculture tree plantations and the reestablishment of native forest both increase carbon sequestration, however the two strategies would likely result in drastically different cobenefits for other environmental goods such as biodiversity (Kanowski *et al.*, 2005). The question then becomes, what incentives do environmental agencies have to increase cobenefits or decrease disbenefits? The goal of this chapter is to shed some light on these incentives.

One potential disincentive that environmental agencies may face has received considerable attention. Adding secondary goals such as increasing cobenefits or decreasing disbenefits might decrease the direct per dollar impact of the agency's spending on the original environmental good they set out to protect. For example, Venter *et al*. (2009) use land-use maps and species distribution maps to model the optimal investment strategy for protecting carbon stocks for the UN's REDD program, given a fixed budget. They then show how this investment pattern would change if there were also an emphasis on increasing the cobenefits of protecting biodiversity. They find that there are direct trade-offs between these two goals, meaning that increasing the biodiversity cobenefits of carbon sequestration efforts would directly decrease the amount of carbon sequestered by the program (given a fixed budget), though they note that due to nonlinearities in the trade-offs, the direct reduction in carbon sequestered would be small at first. They show that these direct trade-offs result from the higher opportunity cost of lands that are rich in biodiversity. In an analysis of multiple ecosystem services in the Willamette Basin, Oregon, Nelson *et al*. (2008) similarly find that there are direct trade-offs between carbon sequestration goals and species conservation goals. Because these direct trade-offs increase the cost per dollar of protecting the original environmental good of interest, they provide a disincentive for increasing cobenefits.

While direct trade-offs have received considerable attention, there may be indirect effects of cobenefits and disbenefits that also influence agencies' incentives. To see this, consider the case of a PES program in Los Negros, Bolivia. Initially, an international environmental agency created a PES program to protect biodiversity, and hoped to eventually collaborate with downstream water-users who would also benefit from upstream forest protection. However, Asquith *et al*. (2008) found that individual water-users benefiting from hydrological services were reluctant to contribute to the program. The authors state the problem concisely: "...using biodiversity payments to pump-prime the [PES] scheme may also have created a perverse incentive for downstream users—why should they pay when someone is already doing it for them?" In this case there was an indirect effect created by the cobenefits of the international agency's conservation efforts. The international agency's protection efforts provided cobenefits of hydrological services.[1] This disincentivized downstream users from investing in environmental protection, which forced the international agency to bear most of the costs.

The impact of these indirect effects on environmental agencies' incentives to provide cobenefits or avoid disbenefits needs to be considered, but has not received significant attention. The goal of this model is to create a basic picture of how these indirect effects might influence the behavior of environmental agencies and what this means for efforts to protect multiple interconnected environmental goods. To accomplish this, first a general model of two environmental goods is developed. Two environmental agencies are assumed to have an interest in protecting these goods. Each of the agencies is tasked with reaching a fixed environmental goal for one of the goods. To examine the incentives agencies face *vis-à-vis* increasing cobenefits and decreasing disbenefits, I show how changes in the agency's cobenefits and disbenefits levels would affect the expenditures required for them to meet their original environmental goal. The model shows that cobenefits and disbenefits may result in strategic behavior on the part of the environmental agencies. In the case of cobenefits this is because one agency has an incentive to free-ride off the other agency's cobenefits. Free-riding is a common problem with public goods and common-pool resources because benefits derived from these goods are non-excludable (Cornes & Sandler, 1996; Sandler, 2004). It is for this reason that strategic behavior is central to the challenges surrounding cobenefits and disbenefits. In particular this model shows that the

[1] This situation is analogous to the classic "chicken game" from game theory. For more information and examples see Gibbons (1992).

existence of cobenefits and disbenefits may results in reciprocal externalities between protection efforts. Considerable work has been done on reciprocal externalities (e.g., foundational work such as Buchanan and Kafoglis (1963), and Vincent (1969)). This model brings existing understanding about public goods, common pool resources and reciprocal externalities to bear on the issue of protecting multiple environmental goods in the presence of cobenefits and disbenefits.

The scope of this model is limited to those environmental goods that are non-exludable (i.e., public goods and common pool resources). For example, this would include environmental goods such as carbon sequestration, biodiversity conservation, and many watershed services, because these have non-excludable benefits such as climate regulation. This also means that the model does not apply to goods such as agricultural commodities, which are private goods. This is a limitation of the model in that I am not considering cases where the goods are characterized by excludability. However, because the inability to exclude users creates many of the environmental challenges targeted by environmental protection efforts, this work is still broadly applicable.

The Model

Suppose there are two environmental goods, the values of which are denoted G^i, for $i = 1,2$, and there are two environmental agencies. The first agency has a conservation goal for one of the environmental goods, and the second agency has a conservation goal for the other environmental good. The agencies' goals can be represented as follows: Agency 1 wants to achieve $G^1 \geq \overline{G^1}$, and agency 2 wants to achieve $G^2 \geq \overline{G^2}$. This allows the superscript to identify both the agency and its environmental goal.

This assumption of a fixed environmental target may make sense for some environmental agencies and not for others. For example, many governments are tasked with reaching set environmental goals (such as a certain water quality threshold or specific reductions in deforestation rates such as those proposed in REDD). In these cases, once the goal is reached, the agency would likely spend additional funds on other initiatives (e.g., other government programs). In the case of environmental NGOs it is possible that they would not set a fixed environmental goal, but would instead use some other criteria to guide the allocation of their funding. Nevertheless, even for environmental NGOs it is possible that once a certain level of environmental protection is achieved (say, successful stabilization of a vulnerable species), it may cease spending money on that initiative, and invest any extra funds on other initiatives that benefit the NGO (e.g., environmental protection in another location, increased fund-raising activities, etc.).

Each agency chooses an investment level, x_i ($i = 1,2$), in order to reach their respective environmental goal at minimum cost. To allow for the possibility of cobenefits and disbenefits, the levels of the two environmental goods are determined by both agencies' investments so that the levels of the two environmental goods are $G^i = f^i(x_i, x_j)$, for $i \neq j, i = 1,2$. Naturally it must be the case that $f^i_{x_i} > 0$ (i.e., an agency's efforts to protect the good that it cares about increase the level of that good), however $f^i_{x_j}$ may be positive or negative to allow for cobenefits or disbenefits. Cobenefits exist when $f^i_{x_j} > 0$ and disbenefits exist when $f^i_{x_j} < 0$ for $i \neq j, i = 1,2$. In order for this problem to be reasonable, it must be the case that $f^i_{x_i} > | f^i_{x_j} |$, $i \neq j, i = 1,2$. This ensures two things. In the case of cobenefits this ensures the environmental agencies choose the most effective conservation spending scheme—it reflects the reasonable assumption that they would not chose an inferior protection strategy if there were a better strategy available. In the case of disbenefits it avoids the case where it would not be possible to simultaneously reach both agencies' environmental protection goals because of the magnitude of the negative externality.

There are several key assumptions contained within these production functions. First, each agencies' investment is assumed to be non-allocatable and non-rivalrous in production. This means that an agency's investment cannot be allocated to one good or another, but rather works toward the production of both. Non-rivalry indicates that the contribution of the investment to one good does not reduce that investment's impact on the other good. These are likely to be reasonable assumptions when the inputs are dollars dedicated to environmental protection by the two agencies. Since agencies are likely to enact conservation actions such as land-use changes, it seems reasonable to assume that dollars spent converting a plot of land to a new land-use could contribute in a non-rivalrous manner to the production of two environmental goods that derive from the new land-use. Another assumption is that the production technologies are non-joint in output (Kohli 1981, 1983), meaning that the level of one good does not depend on the level of the other good. This assumption may be more restrictive. Given the complexity of ecosystems it is likely that the environmental goods themselves may influence one another's production. For example, Diaz *et al*. (2009) found that biodiversity levels influences the long-term storage capacity of carbon stocks. In the context of this model, this would mean that the marginal product of efforts to sequester carbon could be a function of the level of biodiversity, something this specification of production functions does not allow for. A more complete model would account for this possibility, however this would not allow for separable production functions and would significantly complicate the analysis. Instead, as a first step for analyzing this problem, a simpler approach is taken here. Future work should consider this issue.

Each agency wants to achieve its respective environmental target at minimum cost. Because each agency's actions affect both of the environmental goods, each agency must take into account the potential actions of the other agency when making investment decisions. I assume that the two agencies simultaneously make their decisions about how much to invest. It would be possible to model this situation as a sequential game where one agency chooses its investment level first, and the second agency chooses its investment based on the choice of the first agency. Understanding which model (simultaneous vs. sequential) is a better model of reality would require an investigation of the context in which the agencies are acting. The goal of this chapter is to establish a starting point for analyzing environmental protection efforts in the presence of cobenefits and disbenefits, so I will focus on the more tractable case where the two agencies choose their investment levels simultaneously.

For two agencies that choose their investment levels simultaneously, their problems are $min_{x_i \geq 0}\ costs = x_i \quad s.t.\ G^i = f^i(x_i, x_j) \geq \overline{G^i} \quad where \quad i \neq j, i = 1,2$. This implies that there exist equations $x_i(x_j, \overline{G^i})$ for $i \neq j, i = 1,2$, which are the best response functions for the environmental agencies. These functions can take the value of zero (a corner solution), meaning that an agency's best response is to choose zero investment. However, note that even in the case of a corner solution for investment, the environmental good is still greater than zero. To see when corner solutions may arise, note that agency i's investment level tends toward zero (a corner solution) when agency j's environmental goal is larger in size and when the cobenefits of agency j's conservation efforts, $f^i_{x_j}$, are of a larger magnitude. In other words, agency i is more likely to choose to not invest at all if agency j has a large protection goal and produces a high level of cobenefits. This is because agency i's environmental protection goal will be satisfied even if agency i takes no action. For an interior solution where x_i and x_j are positive, each agencies' constraint will bind. Here we can use the implicit function theorem to find $\frac{\partial x_i}{\partial x_j}$ (Simon & Blume, 1994). If we define $H = \overline{G^i} - f^i(x_i, x_j) = 0$, then

$$\frac{\partial x_i}{\partial x_j} = -(\frac{H_{x_j}}{H_{x_i}}) = -(\frac{f^i_{x_j}}{f^i_{x_i}}) \quad for \quad i \neq j, i = 1,2 \tag{1}$$

This indicates that for cobenefits (when $f^i_{x_j} > 0$), $\frac{\partial x_i}{\partial x_j} < 0$, meaning that agency i will reduce its investment level as agency j increases its investment level. Because agency i decreases its protection efforts when agency j increases its efforts, this indicates that in the case of cobenefits, the two agencies' protection efforts are strategic substitutes (Bulow *et al.*, 1985). Alternatively, for disbenefits (when $f^i_{x_j} < 0$), $\frac{\partial x_i}{\partial x_j} > 0$, meaning that agency i will increase its investment level as agency j increases it investment level. In this case, because agency i increases its protection efforts when agency j increase its efforts, the two agencies' efforts are strategic complements (Bulow *et al.*, 1985).

It is important to note how this strategic behavior impacts the additionality of the agencies' protection efforts. (i.e., the additional protection of G^i achieved by a marginal increase in spending by agency i). Using the agencies' best response functions, the levels of the environmental goods can be rewritten as $G^i = f^i(x_i, x_j(x_i)),\ where\ i \neq j, i = 1,2$. Thus the additionality of agency i's protection efforts is,

$$G^i_{x_i} = f^i_{x_i} + f^i_{x_j}\left(\frac{\partial x_j}{\partial x_i}\right) \tag{2}$$

Equation (2) shows how the strategic behavior of the agencies affects the additionality of protection efforts. Recall that $f^i_{x_j}$ represents how agency j's efforts affect G^i and $\frac{\partial x_i}{\partial x_j}$ represents how agency j responds to investment by agency i. This means that the second term in equation (2) represents the strategic feedback due to strategic substitutability or strategic complementarity between the agencies' protection efforts. Thus in the case of cobenefits and disbenefits, strategic behavior results in a decrease in the additionality of protection efforts. To see this, note that in the case where there are cobenefits of protection efforts $f^i_{x_j} > 0$ and $\frac{\partial x_i}{\partial x_j} < 0$. For disbenefits, $f^i_{x_j} < 0$ and $\frac{\partial x_i}{\partial x_j} > 0$, making the second term negative. For use in the next sections substitute equation (1) into equation (2) to yield

$$G^i_{x_i} = f^i_{x_i} - f^i_{x_j}\left(\frac{f^j_{x_i}}{f^j_{x_j}}\right) \tag{3}$$

First focus on the incentives agencies face with respect to increasing the cobenefits of their protection efforts. First assume that there is an interior solution where both environmental agencies choose positive investment levels. Agencies have an ability to chose to what extent they can create cobenefits (e.g., Venter *et al.*, 2009). To examine what incentive agencies have to increase cobenefits, I will show how the costs of reaching the agency's original environmental goal change if the agency undertakes a marginal increase in its level of cobenefits, $f_{x_i}^j$. As noted by Venter *et al*. (2009) and Nelson *et al* (2008), there may be direct costs associated with increasing cobenefits. However, I will assume direct costs to be zero in order to isolate and highlight the indirect costs that result from strategic behavior. The assumption that the agency can enact a marginal increase in its cobenefits with zero direct costs means that I assume that $f_{x_i}^i$ does not change when there is a marginal increase in the cobenefits level, $f_{x_i}^j$. An example helps clarify what this thought experiment might look like in the real world. Suppose there are two carbon sequestration strategies that are equally effective in terms of carbon sequestered per dollar, but which have different impacts on biodiversity—one protects slightly more biodiversity than the other. A marginal increase in the cobenefits ($f_{x_i}^j$) in the sense described above is analogous to switching from the lower biodiversity strategy to the higher biodiversity strategy. The incentives the agency has to make this switch can be seen by looking at the impact of this switch on the additionality of protection efforts and the costs of reaching the agency's original environmental goal.

To see the impact on additionality and costs refer back to equation (3). Equation (3) shows that the additional G^i achieved by a marginal increase in x_i

decreases when there is an increase in agency i's cobenefits $f_{x_i}^j$. This is because when agency i increases its cobenefits, this allows agency j to reduce its spending needed to achieve its own conservation goal. This in turn decreases how much G^i is provided by agency j's cobenefits. As a result, agency i's conservation dollars add less additional G^i, meaning that protection per dollar is lower. Thus if environmental agency i increases it cobenefits, it must also increase it conservation expenditures in order to meet its original fixed environmental goal. To see this more clearly, consider the case of carbon sequestration and biodiversity protection. If an agency endeavoring to meet a carbon sequestration target decides to increase the amount of biodiversity its carbon sequestration efforts protects (i.e., its cobenefits), this may disincentivize individuals who would have otherwise invested in protecting biodiversity. Because their biodiversity efforts would likely have contributed to meeting the carbon sequestration target, this increases the agency's cost of meeting the original carbon target. It is in this sense that reciprocal positive externalities created by cobenefits of conservation efforts may create challenges for protecting multiple services. In the presence of cobenefits, environmental agencies may actually have a disincentive for increasing cobenefits because they can decrease their private cost of achieving their conservation goal by reducing their cobenefits. This is also a consideration for efforts to understand how much it will cost to reach important environmental targets. For example, in the case of REDD, there is great interest in increasing the biodiversity cobenefits. However, it may be necessary to consider how the indirect effects of increasing biodiversity cobenefits will increase the costs to national governments of meeting their carbon sequestration targets.

The impact on additionality itself is worth note given that additionality is generally one of the key criteria in large scale programs directed at carbon sequestration, such as carbon credit schemes (Gustavsson *et al.*, 2000). This model indicates that conservation efforts with higher cobenefits may achieve lower additionality at the margin due to the presence of positive reciprocal externalities. Thus the indirect effects that result from cobenefits should be considered when assessing the additionality of conservation mechanisms such as carbon credits.

Above, each agency was assumed to have chosen a positive level of investment. Now consider the case where agency j's optimal choice is to invest zero (i.e., a corner solution where $x_j = 0$). When agency j finds it optimal to not invest, there are no indirect effects if agency i undertakes a marginal increase in its level of cobenefits. In this case, additionality is simply $f^i_{x_i}$, and agency i can undertake a marginal increase its level of cobenefits without having to increase expenditures to meet its original conservation goal (again this is assuming direct costs are zero). This is because marginal increases in the cobenefits do not trigger reductions in x_j on the part of agency j because it is already at zero investment. Because agency j is pushed toward a corner solution as agency i's environmental goal increases in size and when the cobenefits of agency i's conservation efforts, $f_{x_i}^j$, are larger, these are the conditions under which agency i may be less likely to face disincentives to increase cobenefits from strategic behavior.

The next step is to examine what incentives agencies have to reduce disbenefits of their conservation efforts. Here an interior solution where both environmental agencies choose positive investment levels is guaranteed. As in the case of cobenefits, I assume that there are no direct costs of decreasing disbenefits in order to isolate the impact of indirect costs that result from strategic behavior.

Too see the incentives for decreasing disbenefits, refer to the additionality derived in equation (3). In this equation, agency i's marginal disbenefits are represented by $f_{x_i}^j$. Because $f_{x_i}^j < 0$, by decreasing the magnitude of its disbenefits agency i increases the additionality of it protection efforts. This is because when agency i decreases its disbenefits, this allows agency j to reduce its spending needed to achieve its own conservation goal, which in turn decreases how much G^i is compromised by agency j's protection efforts. This means that if agency i decreases it disbenefits, it will also decreases the level of expenditures required to meet it original conservation goal. Thus in the presence of disbenefits, the environmental agencies actually have an incentive to decrease disbenefits because this decreases their cost of achieving their conservation goal. This is an established result for the case of negative reciprocal externalities (Cornes & Sandler, 1996; Sandler, 2004), but it has a useful interpretation in the case of disbenefits between environmental goods such as ecosystem services. It indicates that the agencies may have some incentive to decrease their negative impact on the other environmental goods because indirect effects can reduce their costs of reaching the own environmental goal.

A Specific Case for Illustration

A specific example helps show these results more clearly. Suppose that the values of the environmental goods are linear in the protection efforts and take the following form $G^i = f^i(x_i, x_j) = x_i + \beta_{ji} x_j$, where $\beta_{ji} \in (-1,1)$ for $i \neq j, i = 1,2$. This allows for the case of cobenefits (when the β terms are greater than zero) and the case of disbenefits (when the β terms are less than zero). To link this to the general model above, note that $f^i_{x_i} = f^j_{x_j} = 1$, $f^i_{x_j} = \beta_{ji}$ and $f^j_{x_i} = \beta_{ij}$. In some cases this assumption of constant returns to scale may be a reasonable assumption, and in others it may not. For example, the value of carbon sequestered per dollar spent might be relatively linear because of the magnitude and spatial scale of the carbon emission problem. However, the values of many environmental goods (or proxies for these goods) accrue in a non-linear fashion (e.g, Barbier *et al*., 2008; Koch *et al.*, 2009). Nevertheless, this assumption simplifies analysis and preserves the basic intuition. Relaxing the assumption of constant returns to scale would change the magnitude of the strategic response, however the signs of the responses would remain the same.

This special case yields the following results: The best response functions are

$$x_i(x_j) = \overline{G^i} - \beta_{ji}x_j \quad for \quad i \neq j, i = 1,2 \tag{4}$$

where

$$\frac{\partial x_i}{\partial x_j} = -(\frac{f^i_{x_j}}{f^i_{x_i}}) = -\beta_{ji} \quad for \quad i \neq j, i = 1,2. \tag{5}$$

From these best response functions, the Nash equilibrium of this problem gives the expenditures required for agency i to reach its conservation goal:

$$x_i^* = \frac{\overline{G^i} - \beta_{ji}\overline{G^j}}{1 - \beta_{ji}\beta_{ij}} \quad for \quad i \neq j, i = 1,2. \tag{6}$$

Finally, from equation (3), additionality for this case is

$$G^i_{x_i} = f^i_{x_i} - f^i_{x_j}(\frac{f^j_{x_i}}{f^j_{x_j}}) = 1 - \beta_{ji}\beta_{ij} \quad for \quad i \neq j, i = 1,2. \tag{7}$$

In the case of cobenefits (when $\beta_{ij}, \beta_{ij} \in (0,1)$), β_{ij} represents agency i's cobenefits. One can see the impact of agency i increasing its cobenefits by looking at the marginal impact of β_{ij} on agency i's expenditures and the additionality of agency i's protection efforts. The disincentive for agency i to increase its cobenefits is clear. Since $\frac{\partial x_i^*}{\partial \beta_{ij}} > 0$, a marginal increase in the agency's cobenefits will increase the cost of obtaining its environmental protection goal, $\overline{G^i}$. This is because the additionality of agency i's conservation efforts (protection achieved per dollar) decreases when it increases its cobenefits, which can be see by the fact that $G^i_{x_i \beta_{ij}} < 0$.

In the case of disbenefits (when $\beta_{ij}, \beta_{ij} \in (-1,0)$), β_{ij} is agency i's disbenefits. As in the case of cobenefits, the impact of agency i decreasing its disbenefits is given by the marginal impact of β_{ij} on agency i's expenditures and the additionality of agency i's protection efforts. In this case agency i has an incentive to decrease its disbenefits. This is because a marginal decrease in the agency's disbenefits will decreases the cost of obtaining its environmental protection goal, $\overline{G^i}$. This can be seen by the fact that $\frac{\partial x_i^*}{\partial \beta_{ij}} < 0$. The agency's costs decrease because the additionality of its protection efforts increases when it decreases its disbenefits, which can be see by the fact that $G^i_{x_i \beta_{ij}} > 0$.

Conclusions

The goal of this model was to examine the incentives that environmental agency might have to increase cobenefits or decrease disbenefits of their conservation efforts. It showed that in the case of cobenefits, an agency that increases it cobenefits may decrease the additionality of their conservation investments and thus increase the costs required to meet their original conservation goal. Given that there are likely direct costs to increasing cobenefits as well (Venter *et al*., 2009; Nelson *et al*., 2008), the additional costs from indirect effects may provide a further disincentive for environmental agencies to provide cobenefits. This conclusion is especially relevant to conclusions drawn by Venter *et al*., (2009) about the prospect for increasing the cobenefits of the UN's REDD program. They claim that because of non-linearities in the direct trade-offs among services, the biodiversity co-benefits of carbon sequestration efforts of REDD could be increased considerably without significant reductions in the amount of carbon sequestered (given a fixed conservation budget). However they only considered direct costs of increasing cobenefits (e.g., increased costs resulting from the need to purchase higher opportunity cost lands that are richer in biodiversity). As this model shows, there might be additional indirect costs due to strategic behavior on the part of other environmental agencies. These costs would further reduce the amount of carbon sequestered per dollar, meaning that the trade-offs between carbon sequestration and species conservation may be higher than they indicated. In addition, given the high priority of additionality in emissions reduction schemes, this model's finding that increasing cobenefits could reduce additionality suggests that this issue may need to be considered in the context of programs like REDD and carbon credit schemes. In the case of disbenefits, the model indicated that reducing the level of disbenefits may actually allow the agency to increase the additionality of their conservation investments and decrease the costs required to meet their original conservation goal. Thus if there are direct costs of decreasing disbenefits, indirect reductions in costs as a result of reducing disbenefits may help offset any direct costs.

While the potential for these incentives ought to be considered, they will certainly not be present in all cases. As shown above, if agency j is in a corner solution where they find it in their best interest to invest zero, the effects illustrated in this model would not be an issue for marginal changes in cobenefits. This is because there would be no strategic behavior resulting from a change in cobenefits given that agency j is already investing zero dollars in environmental protection. If this were the case, an agency's incentives would derive only from the direct costs of increasing cobenefits.

This analysis is also limited by the assumption that the two agencies' efforts will either be cobenefits or disbenefits. It is possible that the two agencies may not have externalities of the same sign. That is, one agency may have cobenefits where the other may have disbenefits. In this sense, the analysis here does not consider the full breadth of situations likely to be faced in reality. However, this model is general enough to consider this case, and so future analysis could focus on how results change if the assumption that externalities are of the same sign is relaxed.

In summary, the value of this model is to illustrate how the nature of the production of environmental goods may result in reciprocal externalities between protection efforts. In the case of cobenefits, the positive reciprocal externalities may cause challenges for achieving successful management of multiple environmental goods in that they provide a disincentive to increase cobenefits. In the case of disbenefits, the negative reciprocal externalities may give agencies an incentive to decrease their negative impact on other environmental goods. These results provide some basic theoretical insights related to the management of multiple environmental goods when there are cobenefits and disbenefits of environmental protection efforts.

Chapter 3

PAYMENTS FOR ECOSYSTEM SERVICES AND POVERTY: TRADE-OFFS BETWEEN ENVIRONMENTAL PROTECTION AND POVERTY REDUCTION

Summary

This work models an environmental agency that would like to design a payments for ecosystem services (PES) program that simultaneously increases environmental protection and achieves development goals (e.g., poverty relief) for the program participants. The primary focus here is to examine inherent trade-offs between these two goals. The model indicates that in theory the trade-offs between these two goals may depend on relevant parameters of the system, particularly the ratio of the price of monitoring to participant's compliance cost. I show that when the ratio of monitoring costs to compliance cost is higher, trade-offs between environmental protection and poverty reduction are likely to be smaller. And when the ratio of monitoring costs to compliance costs is lower, trade-offs are likely to be larger. This analysis is done only for the case of a risk-neutral landowner.

Introduction

There are an increasing number of payments for ecosystem services (PES) programs that are interested in both increasing environmental protection and improving the welfare of the individuals who participate in the program, particularly when these programs are located in less-developed countries and program participants are poor (Grieg-Gran *et al*., 2005; Pagiola *et al*., 2005; Pfaff *et al*., 2007; Wunder, 2008; Tallis *et al*., 2008). However, there is considerable debate surrounding whether or not this dual goal can be achieved and what trade-offs exist between environmental protection and poverty alleviation. (Bulte *et al*., 2008; Landell-Mills & Porras, 2002; Kinzig *et al*., 2011; Corbera & Pascual, 2012; Kinzig *et al*., 2012).

Much theoretical and empirical work has been focused on determining when these goals overlap. Wunder (2008) outlines four ways in which PES can impact the poor: (1) access to PES programs; (2) impacts on sellers of ecosystem services who are poor; (3) impacts on services users/buyers who are poor; and (4) indirect or derived effects such as impacts on food prices in local markets. If the poor are not able to participate in PES programs, this prevents them from receiving direct benefits from the programs unless they themselves benefit from the increase in environmental protection. For example, in Costa Rica's PSA program, national law initially did not allow public funds to be paid to landowners who lacked a formal title even though their land tenure was secure. Since poor people were more likely to lack formal titles than wealthier farmers, this prevented many poor from participating (Pagiola *et al*., 2005). Other barriers to participation include factors such as higher transaction costs for enrolling poor landowners and the potential that poor landowners lack sufficient capital to cover initial costs of participating (e.g., capital costs of afforestation plantations) (Pagiola *et al*., 2005; Pfaff *et al*., 2007).

If the poor are able to gain access to the PES program, a primary issue is how much producer's rent is captured by the poor landowner. If PES programs are in fact voluntary, it is reasonable to assume participants are receiving compensation equal to or greater than the production value they give up, making them at least no worse off than they would be in absence of the program (Zilberman *et al*., 2008), though it is possible that the PES program closes off future options to the landowner such as certain landuses. Indeed, in cases where participation has resulted in reduced income, such as the Sloping Land Conservation Program in China, it has turned out that the individuals involved were actually forced to participate (Bennett, 2008). The question then becomes how much economic rent is captured by the program participant. Wunder (2008) points out that "as in any commercial transaction, there is an inherent conflict over price between ES buyers maximizing consumer surplus ('biggest conservation bang for the buck') and ES providers boosting their provider surplus (PES minus opportunity costs)." Information asymmetry is a key factor in determining how much economic rent poor participants are likely to capture. For example poor participants are likely to have disproportionate information about their compliance costs (relative to buyers). However, Pagiola *et al*. (2005) note that in some cases, the opportunity costs of upstream providers is likely easier to calculate than downstream user's willingness to pay, which would reduce information rents by placing the producers in an inferior negotiating position.

Information asymmetry also arises if there is imperfect monitoring of program participants. Thus participants may stand to benefit if the agency is unable to perfectly monitor compliance with the terms of the PES contract. Sellers may also receive non-pecuniary benefits such as increases in human capital from training programs (Kerr, 2002; Grieg-Gran *et al*., 2005) and increases in land tenure security (Miranda *et al*., 2003; Robertson & Wunder, 2005).

If the poor are the service buyers (i.e., those paying for the service), there is a risk that they will now be required to pay for benefits that previously were free. While this concern needs to be considered in each case, there are several reasons why those buying services are less likely to be poor: In practice many buyers are actually powerful monopsonies or oligopsonies (e.g., hydroelectric power companies); the poor generally hold less-developed land that is more strategic for service provision; several ecosystem services are considered luxuries (e.g., scenic beauty); and groups of poor potential buyers often do not have the coordination necessary to organize a payment scheme (Wunder, 2008). Nonetheless, there is still potential for costs to be passed on to poor users (e.g., prices for services that previously were free, increases in water prices in urban areas, etc.).

Finally, indirect effects of conservation incentive programs on the poor are also possible. For example, land-use change such as afforestation or the transition of agricultural land back to natural habitat are likely to impact labor markets (positively or negatively) that provide jobs for the poor (Zilberman *et al*., 2008). In addition, increased vigilance and land-tenure security may impact the landless poor’s access to resources such as non-timber forest product (Engel *et al*., 2005; Kerr, 2002).

The work thus far on the potential for overlap between environmental and poverty reduction goals has provided some useful insight. It is clear that PES is not a silver bullet for reducing poverty. However, in light of the findings above, it is also clear that steps can be taken so that a PES program might be more likely to reduce poverty and avoid negative impacts on the poor. For example, Pagiola *et al*. (2005) notes that it is possible to design PES programs so that they do not exclude the poor, have positive effects on local labor markets (i.e., create jobs), and provide technical assistance or credit when required. However, these initiatives will almost certainly come at a cost, and would thus reduce the efficiency of achieving environmental protection. This highlights the need for further study of the trade-offs between environmental protection and poverty reduction in the context of PES. In particular this indicates a need for understanding how making PES programs more ’pro-poor’ will affect the efficiency of environmental protection effort. The theoretical model in this chapter is an attempt to fill this gap. I show how the behavior of the environmental agency might change when it increases its poverty reduction target, and how this might affect the costs required to meet a fixed environmental target.

To do this, I build on work that models conservation incentive programs in the agri-environmental literature (representative examples include Choe & Frazer, 1998; Choe & Frazer, 1998; Hart & Latacz-Lohmann, 2005). These papers use a principal-agent framework to model conservation incentive programs designed to increase the production of environmental goods from agricultural landscapes. Examples of these programs include The English Countryside Stewardship Scheme, the French *prime à l'herbe* program and the German MEKA program. In most cases agri-environmental programs provide a financial incentive to farmers to undertake some costly restriction of production on their land. In this regard, the programs take a structure quite similar to PES. In the agri-environmental literature, many papers model a conservation agency endeavoring to achieve some fixed level of environmental protection at lowest cost. I build on this approach by adding an explicit goal of also increasing the welfare of the participating landowner. By increasing this poverty reduction goal, it is possible to examine how this changes the environmental agency's optimal behavior and the costs required to achieve its environmental goal.[2]

As noted above, PES can impact the welfare of the poor in many ways. This model focuses only on the pecuniary impacts on the participant's welfare. This means that non-pecuniary impacts on participants (such as increases in human capital) and impacts on non-participants are not considered. Naturally these factors are also important in understanding the trade-offs between environmental protection and poverty reduction, however to make this analysis tractable, it is useful to focus on one dimension of the poverty impacts of PES. In addition, the pecuniary benefit from the incentive payment (net of compliance

[2] Relatively few papers in the PES literature bridge the gap between the PES literature and the agri-environmental literature (exceptions include Ferraro, 2008, Zabel & Roe, 2009), so a secondary contribution of this work is to help bring useful modeling techniques and results from the agri-environmental literature to the PES literature.

costs) is likely to be one of the primary impacts on the welfare of the program participant.

The Model

Suppose an environmental agency wants to contract landowners to increase environmental protection. The landowners are currently engaging in an activity (e.g., agriculture, cattle grazing) which they will have to reduce (at a cost) in order to increase environmental protection. Thus the agency wants to design an appropriate contract to induce the landowners to undertake the costly reduction in their current activity. However the environmental agency would also like to use the program as a poverty reduction mechanism, meaning that it wants the program to simultaneously increase environmental protection and increase the welfare of program participants. Participants are assumed to be homogeneous and all below the poverty line. For example, this might be the case if the program is being implemented in a location where most landowners are poor. I assume that landowners do not derive direct benefits from the environmental improvements (or these benefits are negligibly small), which is likely to the case with many PES programs such as those focused on sequestering carbon to mitigate climate change.

In order to achieve these goals, the agency creates the following contract: The contract stipulates that the landowners must dedicate a set effort level, e, which increases environmental protection (e.g., more biodiversity protection, less run-off, etc.). If the landowner complies and supplies this effort, the landowner will receive a transfer payment, t. However, if the landowner is detected putting forth an effort level lower than e, she will not receive the payment t.

Assume for simplicity that the landowner has only two possible effort levels. She supplies either e or no effort at all. As noted above, supplying effort decreases the land owner's profits from outside activities (e.g., agriculture, raising cattle). For the sake of simplicity, I assume that landowners are homogeneous with respect to their outside earning opportunities and their compliance costs. In reality this is not likely to be the case, and many papers examine the case of heterogeneity of compliance costs (and related issues of adverse selection) (for examples see, Ferraro, 2008; Moxey *et al*., 1999; Wu & Babcock 1996). However, assuming homogeneous landowners permits the convenience of modeling a contract with a representative landowner, which is the approach taken here. If the landowner does not participate, she receives π_o, her reservation wage. And if the agent participates and supplies the required effort, e, her profit from outside activities is reduced to $\underline{\pi}$, where $\underline{\pi} < \pi_o$. Thus her compliance cost for the program is $c = \pi_o - \underline{\pi}$. I assume that this compliance cost is known to the environmental agency, meaning that there are no problems of adverse selection.[3] That is, the landowner cannot hid her compliance costs in order to request a higher compensation payment. In the real world it is certainly possible that compliance costs would not be known, as indicated in Ferraro (2008). However, Pagiola *et al*. (2005), claim that because PES programs usually involve a change

[3] Naturally asymmetric information about compliance costs is another way that landowners can increase the amount of producer's rent they capture. In the future it could be fruitful to do a joint model that focuses on both adverse selection and moral hazard. However for now I focus only on information rents captured by the landowners as a result of imperfect information about actual landowner compliance.

in land-use (e.g., leaving land fallow), it is often relatively easy to calculate the opportunity cost to poor landowners of participating in the program.[4]

This model assumes that the incentive payments offered by the program are fixed payments (e.g., a fixed payment per acre enrolled in the program, etc.). Since landowners are assumed to be homogeneous with respect to landholdings and compliance costs, the total payment size is identical across landowners. This is because the agency chooses a single payment size that is optimal for inducing compliance by all landowners in the program. Fixed payments are the standard in PES programs (Pattanayak *et al*., 2010) and agri-environmental programs (Hart & Latacz-Lohmann, 2005), though in real programs the total payment size may vary across landowners. This could be the case, for example, if one landowner enrolls more land in the program than other landowners.

This model also assumes that there are no fines levied on a landowner who violates the terms of the contract. Fines are not an uncommon feature of conservation incentive programs in the developed world and models in the agri-environmental literature frequently include them as one of the available tools for inducing compliance (e.g., Ozanne, Hogan & Colman, 2001; Hart & Latacz-Lohmann, 2005). However, in this model I assume that the environmental agency does not have this option. This is because the landowners in this model are assumed to be poor, and it is probably politically unacceptable to levy a fine on individuals who are already poor. This is consistent with the design of many PES

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[4] Note that I assume there is no uncertainty associated with the landowner's outside profit earning activities. In reality there is likely to be uncertainty associated with these earnings, especially because these will probably include agricultural activities where weather and crop prices create uncertainty. In the risk neutral case, this assumption is essentially unimportant because π_o can just be considered the expected earnings from outside activities. However, if risk aversion were introduced, this assumption would require more attention.

programs in developing countries (Grieg-Gran *et al.*, 2005; Pattanayak *et al.*, 2010). In the absence of a fine, the only punishment available to the environmental agency is to withhold the payment if the landowner does not supply effort. Because there are only two effort levels, e and zero effort, this means that the worst off the non-complying landowner can be is back to her reservation wage, π_o.

It is worth noting that in some cases, it may not even be politically feasible to require the landowner to return the incentive payment. In these cases a more realistic model would have as the penalty for non-compliance the lost gains of future participation in the program. Indeed this is in line with many PES programs today where individuals found in non-compliance are restricted from participating in the program in the future, or are restricted from reentering the program for a certain number of years (Pattanayak *et al.*, 2010). This dynamic approach is more consistent with models such as those found in the efficiency wages literature (e.g., Shapiro & Stiglitz, 1984). However, for now I maintain a static model for the sake of simplicity.

Naturally the landowner has an incentive to participate in the program but not supply the costly effort. As a result the agency monitors in order to detect non-compliance (e.g., they monitor the landowner's activities, the level of the environmental good, etc.). The probability of the landowner being found in non-compliance depends on whether or not she supplies effort. If the landowner puts forth full effort, e, she will never be found in non-compliance. This means that if she complies with the contract, she will receive the payment t with certainty.

However, if the landowner violates the contract and does not increase environmental protection (i.e., shirks), the principal can only detect this violation with some probability, q, where $q \in [0,1]$. The accuracy of detecting contract violations, q, depends on how much the agency invests in monitoring activities. In this sense, q can be interpreted as the level of monitoring effort, where $q = 0$ is no monitoring effort and $q = 1$ is the level of monitoring effort needed to detect contract violations with one-hundred percent accuracy. As is often done in the agri-environmental literature, I make two simplifying assumptions regarding the accuracy of detecting contract violations: (1) The agency can directly choose the accuracy of detecting contract violations, meaning that it directly chooses q; and (2) expenditures increase proportionately in the accuracy level chosen (Choe & Frazer, 1998; Choe & Frazer, 1999; Ozanne, Hogan & Colman 2001; Hart & Latacz-Lohmann, 2005). Suppose that the cost to the environmental agency of monitoring a landowner is $p_q q$, so that p_q is the price of full monitoring.

The assumption that a complying landowner will receive the payment, t, with certainty reflects the fact that the burden of proof for identifying compliance/non-compliance is most appropriately placed on the environmental agency when the landowners are poor. Other authors modeling agri-environmental programs have taken different approaches. For example, Choe and Frazer (1998, 1999) assume that the environmental agency's imperfect information results in an inability to identify both non-compliance and compliance. That is, the environmental agency identifies the landowner's behavior correctly with some probability. However this implies that even if the landowner complies with the contract there is a positive probability that she will be found in non-compliance. Ozanne, Hogan and Colman (2001) suggest that this may not be an appropriate way to characterize imperfect monitoring, and that a more realistic approach is one where the burden of proof rests on the environmental agency. To capture this, they characterize imperfect monitoring so that the monitoring process fails to detect all landowners who do not comply, but landowners who fulfill the terms of the contract are never found to be in non-compliance. I assume that this latter characterization is more appropriate for PES programs where participants are poor because political norms are likely to be strongly against inadvertently punishing poor participants who have complied with their contracts.

In this model I will assume that the landowner is risk-neutral. Thus the utility for the risk-neutral landowner who chooses not to participate is $U^{decline} = \pi_o$ and the utility for the landowner who chooses to participate and supply effort can be represented as $U^{comply} = t + \pi_o - c$. Her expected utility if she chooses to participate but supply no effort is $U^{shirk} = (1-q)(t+\pi_o) + q\pi_o$.

Because the environmental agency would also like to use the program as a poverty reduction mechanism, the agency also wants those individuals who participate and comply with the program's objective to receive welfare gains from their participation. To formalize this, suppose that the agency has a poverty reduction goal for the landowner that is represented by $U^{comply} \geq \overline{U} = \overline{\pi}$, where

$\overline{U} = \overline{\pi} \geq U^{decline}$. Note that because these are weak inequalities, this allows for a case where the agency has no poverty reduction goal (i.e., when $\overline{\pi} = \pi_o$).

The agency then is faced with the task of how to choose the incentive payment level and monitoring level in order to induce the landowner to supply effort, while simultaneously achieving its poverty reduction goal. Below I set up the agency's problem for a risk-neutral landowner. From this problem one can derive several results, including the optimal incentive payment and monitoring level, the expenditures required to achieve the desired environmental protection, the shadow cost of poverty reduction, and the producer's rents.

As stated above, the environmental agency's goal is to minimize expenditures while meeting its environmental constraint and poverty reduction constraint. Thus its problem is,

$$\min_{t,q} \quad E = t + p_q q \tag{8}$$

$$s.t. \quad P.C. \quad t + \pi_o - c \geq \pi_o$$

$$P.R. \quad t + \pi_o - c \geq \overline{\pi}$$

$$I.C. \quad t + \pi_o - c \geq (1-q)(t + \pi_o) + q\pi_o$$

The structure of this model captures the agency's interest in achieving multiple policy objectives. While its primary goal is to minimize costs, it must do so while also satisfying several constraints that represent its multiple policy objectives. The first and third constraint represent the environmental policy objective. The first constraint is a participation constraint (P.C.), which requires that the payoff to the landowner exceed her compliance costs. This ensures that the landowner would actually participate in the program. The third constraint is the incentive compatibility constraint (I.C.). This requires the payoff of participating and supplying effort to be at least as high as the payoff of participating and shirking. It ensures that the participating landowner supplies the environmental benefit. The second constraint, the “poverty reduction constraint” (P.R.), represents the agency’s poverty reduction objective. It ensures that the complying landowner is made better off by the program. In order to solve this problem, first note that because $\bar{\pi} \geq \pi_o$, if the P.R. is satisfied, the P.C. is always satisfied, meaning that the P.C. can be ignored. Indeed if $\bar{\pi} = \pi_o$ (i.e., the agency has no poverty reduction goal), then the P.R. constraint is identical to the P.C. The optimal levels of t and q can be found by solving the Lagrangian below.

$$L = t + p_q q + \lambda[\bar{\pi} - t - \pi_o + c] + \mu[(1-q)(t+\pi_o) + q\pi_o - t - \pi_o + c] \tag{9}$$

The Kuhn-Tucker first order conditions are,

$$L_t = 1 - \lambda - \mu q = 0$$

$$L_q = p_q - \mu t = 0$$

$$L_\lambda = \bar{\pi} - t - \pi_o + c \leq 0, \qquad \lambda \geq 0, \qquad L_\lambda \lambda = 0$$

$$L_\mu = -qt + c \leq 0, \qquad \mu \geq 0, \qquad L_\mu \mu = 0$$

These conditions provide several results that will be useful for examining the questions put forth at the beginning of the chapter. First, note that the I.C. constraint will always bind at the solution because it is never cost efficient to use more incentive payment or monitoring than necessary to induce compliance. However the P.R. may or may not bind because the information rents captured by the landowner may exceed the poverty reduction goal. Whether or not the P.R. constraint binds depends on the parameters of the problem. As a result, it is convenient to denote when the P.R. constraint is binding and when it is non-binding. To do this define s to be a vector of parameters that is ($p_q, c, \pi_o, \bar{\pi}$) and S to be the set of all possible parameter vectors. Now define S' as the set of parameter vectors for which the P.R. constraint will be binding. Thus the P.R. constraint is binding when $s \in S'$, and the P.R. constraint is non-binding when $s \notin S'$. Using this, the optimal incentive payment and monitoring level are

$$t^*(p_q, c, \bar{\pi}, \pi_o) = \begin{cases} \bar{\pi} - \pi_o + c & \text{if } s \in S' \text{ (P.R. binding)} \\ (p_q c)^{1/2} & \text{if } s \notin S' \text{ (P.R. non-binding)} \end{cases} \tag{10}$$

$$q^*(p_q, c, \bar{\pi}, \pi_o) = \begin{cases} c(\bar{\pi} - \pi_o + c)^{-1} & \text{if } s \in S' \text{ (P.R. binding)} \\ (p_q^{-1} c)^{1/2} & \text{if } s \notin S' \text{ (P.R. non-binding)} \end{cases} \tag{11}$$

And the expenditures required to reach the environmental goal and poverty reduction goal simultaneously are,

$$E(p_q,c,\bar{\pi},\pi_o)=\begin{cases}\bar{\pi}-\pi_o+c+p_q c(\bar{\pi}-\pi_o+c)^{-1} & \text{if} s\in S' \text{ (P.R.binding)}\\[2ex] 2(p_q c)^{1/2} & \text{if} s\notin S' \text{ (P.R.non}-\text{binding)}\end{cases} \qquad (12)$$

The Lagrangian multiplier of the P.R. constraint, λ, has a useful interpretation. It is the shadow cost of poverty reduction, which shows the marginal cost to the agency of increasing its poverty reduction goal while still achieving its environmental goal. Because such increases in expenditures would result in less money available for spending on conservation (e.g., less money for enrolling more landowners in the program), λ indicates the degree of trade-offs between environmental protection and poverty reduction. The value of λ is,

$$\lambda(p_q,c,\bar{\pi},\pi_o)=\begin{cases}1-p_q c(\bar{\pi}-\pi_o+c)^{-2} & \text{if} s\in S' \text{ (P.R.binding)}\\[2ex] 0 & \text{if} s\notin S' \text{ (P.R.non}-\text{binding)}\end{cases} \qquad (13)$$

Finally, because we are interested in the welfare of the landowner, another useful result is the producer’s rent, which I’ll denote B. It is,

$$B(p_q, c, \bar{\pi}, \pi_o) = \begin{cases} \bar{\pi} - \pi_o & \text{if } s \in S' \text{ (P.R.binding)} \\ (p_q c)^{1/2} - c & \text{if } s \notin S' \text{ (P.R.non-binding)} \end{cases} \quad (14)$$

Recall that the goal at hand is to see how a poverty reduction constraint will affect the optimal behavior of the environmental agency and to see how this translates into trade-offs between environmental protection and poverty reduction. Given this goal, a useful heuristic is to begin with the case of an agency that has no poverty reduction goal, and then examine the impacts of the first marginal increase in poverty reduction. To do this we can start by evaluating the results above for $\bar{\pi} \leq \pi_o$. This provides the results of the agency's problem if it is seeking only to achieve the environmental goal (by inducing compliance) at lowest cost. This can be used to determine how the first marginal increase in the poverty reduction will affect the environmental agency's cost of achieving its environmental goal, and when there may be trade-offs between environmental protection and poverty reduction. It is useful to start with the first-best scenario as a point of reference.

First-best Scenario

Consider the first-best scenario for an agency with no poverty reduction goal. In this case there is no information asymmetry (alternatively this can be viewed as monitoring being costless). As a result, the incentive compatibility constraint collapses to the participation constraint, meaning that the agency simply chooses the minimum level of incentive payment so that the P.R. constraint is satisfied. This means that in the first-best scenario, the P.R. constraint will always be binding. This yields $t^*(\bar{\pi} = \pi_o) = c$, $q^*(\bar{\pi} = \pi_o) = 1$, $E(\bar{\pi} = \pi_o) = c$, $\lambda(\bar{\pi} = \pi_o) = 1$ and $B = 0$. Because the landowner receives zero producer's rent in this case, if the agency wants the program to increase the landowner's welfare, it must increase the size of the incentive payment. This can be seen by the fact that $t^*_{\bar{\pi}} = 1 > 0$. However, because $t = c$ and $q = 1$ represent the efficient incentive payment and monitoring level for achieving environmental protection, this means that when the P.R. reduction goal increases, this causes the environmental agency to depart from the efficient equilibrium for achieving only environmental protection. This departure from the efficient levels of t and q means that any effort to increase the landowner's rent will come at a cost to the environmental agency, indicating that there are trade-offs between the two goals. Another way to view this is that increases in the poverty reduction goal cause a decrease in the amount of environmental protection achieved per dollar. This can be seen by the fact that the shadow cost of the first marginal increase in poverty reduction is positive. Indeed, because $\lambda = 1$ in the first-best scenario, the cost of increasing the amount of poverty reduction achieved by the program is essentially the same as a program that simply provides financial handouts to the landowner without tying them to the provision of an environmental good. In other words in the first-best scenario, a dollar increase in the welfare of the program participant (beyond their reservation wage) is a foregone dollar that could have been spent on more environmental protection by enrolling more landowners in the program.

Second-best Scenario

In most cases, however, monitoring is likely to come at some cost to the environmental agency (Choe & Frazier, 1998, 1999; Hart & Latacz-Lohmann, 2005). Therefore, one must consider the trade-offs between environmental protection and poverty reduction in the context of a second-best world where information about compliance is costly, and there exists moral hazard. This is a relevant case because moral hazard may allow the landowner to capture information rents. In the second-best case, where there is a positive price of monitoring, it is no longer clear whether or not the P.R. constraint will bind. Because the results above depend on whether or not the P.R. constraint is binding or non-binding, it is useful to determine under what conditions the P.R. constraint is binding ($s \in S'$) and non-binding ($s \notin S'$). If we denote t^{*NB} as the optimal incentive payment when the P.R. constraint is non-binding, then the P.R. constraint will be binding whenever $t^{*NB} < \bar{\pi} - \pi_o + c$, that is when $(p_q c)^{1/2} < \bar{\pi} - \pi_o + c$. Thus when the environmental agency has no poverty alleviation goal (when $\bar{\pi} = \pi_o$), this condition states that the P.R. constraint is binding when $p_q < c$.

First consider the second-best scenario for an agency with no poverty reduction goal where the P.R. is binding, i.e., when $p_q < c$. Here the results are as follows: $t^*(\bar{\pi} = \pi_o) = c$, $q^*(\bar{\pi} = \pi_o) = 1$, $E(\bar{\pi} = \pi_o) = c + p_q$, $\lambda(\bar{\pi} = \pi_o) = 1 - \frac{p_q}{c}$ and $B = 0$. In this case the landowner still does not capture any producer rent when the agency has no poverty reduction goal. As a result, if the agency wants the program to increase the landowner's welfare, it must alter the levels of incentive payment and monitoring. This can be seen by the fact that $t^*_{\bar{\pi}} > 0$ and $q^*_{\bar{\pi}} < 0$. This means that when $p_q < c$, the first marginal increase in the poverty reduction goal causes the agency to increase the size of the financial incentive and decrease monitoring. However, as in the first-best scenario, this represents a departure from the efficient levels of incentive payment and monitoring for achieving just the environmental goal at lowest cost. As a result, the shadow cost of the first marginal increase in poverty reduction is also greater than zero, indicating trade-offs between environmental protection and poverty reduction. However, note that the shadow cost of poverty reduction is less than in the first-best scenario, meaning that when there is a positive price of monitoring and $p_q < c$, the trade-offs are smaller than in the first-best scenario, and that the trade-offs decrease as the ratio of the price of monitoring to compliance costs increases.

Now consider the second-best scenario for an agency with no poverty reduction goal where the P.R. is non-binding, when $p_q > c$. Here the results are as follows: $t^*(\bar{\pi} = \pi_o) = (p_q c)^{1/2}$, $q^*(\bar{\pi} = \pi_o) = (p_q^{-1} c)^{1/2}$, $E(\bar{\pi} = \pi_o) = 2(p_q c)^{1/2}$, $\lambda(\bar{\pi} = \pi_o) = 0$ and $B = (p_q c)^{1/2} - c$. Here the landowner does capture some producer's rent even though the agency has no poverty reduction goal. As a result, for the first marginal increase in the poverty reduction goal, the agency need not alter its incentive payment and monitoring levels from the levels that are efficient for obtaining environmental protection at lowest cost. This can be seen by the fact that $t^*_{\bar{\pi}} = 0$ and $q^*_{\bar{\pi}} = 0$ in this case. As a result the shadow cost of the first marginal increase in the poverty reduction goal is zero. This apparent lack of trade-offs is due to the fact that the efficient behavior for meeting the environmental goal at lowest costs already produces positive producer's rents. Note that this result is only for the first marginal increase in the poverty reduction goal (above zero). As the magnitude of $\bar{\pi}$ increases, the P.R. constraint may eventually bind, resulting in a positive shadow cost of further poverty reduction.

Summarizing these results, we can see a relationship between the shadow cost of the first marginal increase in the poverty reduction goal (an indication of the trade-offs) and the ratio of the price of monitoring to the compliance costs. In the first-best scenario, the ratio of p_q to c is zero (because monitoring is costless) and $\lambda(\bar{\pi} = \pi_o) = 1$. In the second-best scenario where $0 < p_q < c$, it was shown that λ is between zero and one. Finally, in the second-best scenario where where $p_q > c$, it was shown that $\lambda(\bar{\pi} = \pi_o) = 0$. This suggests that the initial trade-offs between environmental protection and poverty reduction are highest when the ratio of p_q to c is lowest, and that the trade-offs are lowest when the ratio of p_q to c is highest. The reasoning behind this result is as follows. As the price of monitoring increases, the agency prefers to use higher incentive payments and lower monitoring levels to induce compliance. This results in higher information rents captured by the landowner. However, if compliance costs are higher, the landowner has a higher incentive to shirk, which in turn increases the value of information (about compliance) to the environmental agency. As a result, it is the ratio of these two parameters that is important in understanding the initial trade-offs between environmental protection and poverty reduction.

Conclusions

These results suggest that there are situations where environmental protection efforts may be more conducive to including a dual goal of poverty reduction, and others situations that may be less conducive to this dual goal. The model indicates that this is linked to the ratio of the price of monitoring, p_q, to the compliance costs, c. When the price of monitoring is high relative to the landowner's compliance costs, trade-offs between environmental protection and poverty reduction are likely to be smallest. And when the price of monitoring is small relative to the landowner's compliance costs, the trade-offs are likely to be largest.

A relevant question is which parameter values exist in the real world, and thus which level of trade-offs are environmental agencies are likely to face. Unfortunately, in empirical work on PES, these parameter values are typically not reported (Grieg-Gran *et al.*, 2005). While empirical information about these parameter values would be ideal, it seems reasonable that there would be considerable variation in these parameters across different PES programs in the real world. Consider first the price of monitoring. For some environmental protection efforts, monitoring may be relatively inexpensive. For example, if the incentive program's goal is to maintain forested lands to secure carbon sequestration, remote sensing can be used relatively effectively to track whether participants have maintained their forest cover (though the issues of enforcement costs and verification that landowners are to blame for losses would also play a role here, and this should be considered). In contrast, some other conservation efforts such as those that target specific endangered species may have higher monitoring costs (e.g., a need for on-the-ground surveying of the species population by trained biologists). It is also likely that compliance costs would vary depending on factors such as the requirements of the contract, the productivity of local soil conditions, etc. Naturally, even given variation in these parameters, the correlation between the price of monitoring and the compliance costs would be important to know. However, with the sparse attention they have received in the empirical literature, this cannot be determined.

While information about real world parameters is scarce, these results still provide some insight about how changes in the capacity of agencies to monitor would affect the trade-offs between environmental protection and poverty reduction. For example, Pattanayak *et al*. (2008) note that the price of monitoring in carbon sequestration schemes is decreasing as remote sensing technologies are becoming more effective. This would suggest that for PES programs designed to increase carbon storage, the trade-offs may become higher over time as monitor capacity improves. This is especially interesting because efforts to sequester carbon represent one of the largest (if not the largest) source of funding for conservation incentive programs in the developing world. Indeed, the UN's proposed REDD program, which is likely to be implemented in the form of many "PES-like" projects (Pattanayak *et al*., 2008), would involve an unprecedented transfer of conservation dollars to developing countries (Venter *et al*. 2009).

This model also suggests that our growing understanding about ecosystem service production functions may be a relevant consideration for assessing trade-offs. In this model the parameter p_q is very general, however many factors contribute to the price of monitoring. If the PES program verifies compliance by monitoring the output of the environmental good (as is suggested by Roe & Zabel, 2009), one of the factors likely to influence the price of monitoring is the level of understanding the environmental agency has about the ecosystem service production function. If the agency has a poor understanding of the production function, it is likely that it will be more difficult (and thus more costly) to detect contract violations. On the other hand if the agency has a better understanding of the production function, this would likely decrease the price of monitoring because it would be easier to detect a violation. This suggests that as knowledge about ecosystem service production functions increases, this may result in higher trade-offs between environmental protection and poverty reduction.

In conclusion, the results from this model are quite limited in that PES programs have the potential to impact the welfare of the poor in a variety of different ways; this model focuses only on the income gains that result from the incentive payment. Nonetheless, the model points toward some areas where it may be good to focus attention in determining the trade-offs between environmental protection and poverty alleviation in the context of a PES program. It also suggests a need for greater attention to measuring relevant parameters such as the price of monitoring and compliance costs as these may be useful in empirical assessments of trade-offs for real-world programs.

Chapter 4

CONCLUSIONS

The context in which environmental protection efforts take place is complex, both biophysically and socially. This complexity often means that multiple policy goals are unavoidably linked. It would be simpler if environmental protection efforts could focus on one problem at a time. However, biophysical, social and political contexts rarely make that feasible. This thesis focused on two such cases that are receiving considerable attention in the literature: (1) the desire to manage multiple environmental goods simultaneously and (2) the desire to use payments for ecosystem services programs to alleviate poverty. The goal of the thesis was to provide some general insights about which situations might be most conducive to achieving multiple policy objectives and in which situations there are likely to be trade-offs.

Chapter two addressed the issue of cobenefits and disbenefits produced by environmental protection efforts. There is much work directed at identifying where there are cobenefits and disbenefits, but little work examining the incentives environmental agencies actually have to provide them. The basic accomplishment of this model was that it highlights how the nature of the production processes of environmental goods may result in reciprocal externalities between environmental protection efforts. These externalities represent social feedbacks that affect the incentives environmental agencies face *vis-à-vis* increasing cobenefits and decreasing disbenefits. In the case of cobenefits, the model indicated that if agencies increase their cobenefits, this may decrease the environmental protection achieved per dollar, making it more costly to reach their original environmental goal. Many mapping exercises have illustrated how it may be possible to undertake protection efforts that achieve larger cobenefits, however whether agencies will actually choose such protection strategies needs to be considered in light of their incentives to do so. This model suggests that theoretically they may face some disincentives to take such a path. In the case of disbenefits, the model indicated that agencies may in fact have incentives to decrease disbenefits because this could increase the environmental protection achieved per dollar and reduce the cost of achieving their original environmental goal. In summary, the direct cost of altering cobenefits and disbenefits levels have received the majority of the attention in the discourse surrounding cobenefits and disbenefits. This chapter argued that indirect effects that result from reciprocal externalities should also be taken into consideration when managing for multiple environmental goods in the presence of cobenefits and disbenefits.

The model in the third chapter was directed at the current interest in using PES as a poverty alleviation mechanism. The model illustrated a PES program where the agency running the program also had a poverty reduction goal. This allowed the trade-offs between the two goals to be explicitly examined by deriving the shadow cost of poverty reduction. The model showed that the initial trade-offs between the two goals hinges on the ratio of the price of monitoring to the compliance cost. These results must be qualified by the fact that the welfare of the landowner is likely to be influenced by a wide variety of program-related impacts (e.g., increased land-tenure security, increased human capital from capacity training, etc.). Nonetheless, the income transfer is likely to be a large component of the welfare impacts a PES program. By explicitly including both of these goals, this model provides a starting point for theoretical models examining trade-offs between the goals of environmental protection and poverty reduction in the context of a PES program.

The theoretical models in this thesis generate hypotheses that future empirical work could test. In the second chapter, the model indicates that agencies may have an incentive to decrease cobenefits and reduce disbenefits. This results in two hypotheses. In the presence of cobenefits (i.e., when there are positive reciprocal externalities between environmental protection efforts), agencies will reduce the degree to which their conservation investments supply cobenefits. The second hypothesis from this chapter is that in the presence of disbenefits (i.e., when there are negative reciprocal externalities between environmental protection efforts), agencies will reduce the degree to which their conservation investments produce disbenefits.

In the chapter three, the model indicates that the trade-offs between environmental protection and poverty reduction depend on the parameters of the system, particularly the price of monitoring, the compliance cost and the size of the poverty reduction goal. One particularly relevant hypothesis that these results generate is that as the price of monitoring increases, the trade-offs between environmental protection and poverty reduction will decrease (assuming other parameter values are held constant). In order to carry out an empirical test of this hypothesis, it could be fruitful to examine multiple projects within one agency that vary in the price of monitoring and their impact on participant welfare, but which have relatively similar compliance costs. In this way one could examine whether changes in the price of monitoring affect the degree to which participant welfare is increased by the program.

In both chapters, the models indicate potential challenges to achieving additional policy goals on top of the environmental agency's original environmental protection goal. In chapter two, increasing cobenefits resulted in higher costs of reaching a fixed environmental goal. In chapter three, increasing the poverty reduction goal often came at a cost to the environmental agency. However these two chapters also showed that in some cases, environmental agency might be more likely to achieve multiple policy goals. In chapter two this was the case where the agencies had an incentive to decrease disbenefits; and in chapter three this was the case where parameter values were such that the efficient protection of the environmental good resulted in positive rent for the landowner, even in the absence of a poverty reduction goal. In conclusion, these models do not provide any specific policy advice; they are simplistic and too general. But for the two cases presented in chapter two and three, they do provide some general guidance for where attention might be focused as we assess the challenges of achieving the multiple policy goals that arise from the context in which environmental protection efforts take place.

REFERENCES

Anderson, B. J., Armsworth, P. R., Eigenbrod, F., Thomas, C. D., Gillings, S., Heinemeyer, A., Roy, D. B., et al. (2009). Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology*, *46*(4), 888–896. doi:10.1111/j.1365-2664.2009.01666.x

Baker, J. P., Hulse, D. W., Gregory, S. V., White, D., Van Sickle, J., Berger, P. A., Dole, D., et al. (2004). Alternative futures for the Willamette River Basin, Oregon. *Ecological Applications*, *14*(2), 313–324. doi:10.1890/02-5011

Barbier, E. B., Koch, E. W., Silliman, B. R., Hacker, S. D., Wolanski, E., Primavera, J., Granek, E. F., et al. (2008). Coastal Ecosystem-Based Management with Nonlinear Ecological Functions and Values. *Science*, *319*(5861), 321–323. doi:10.1126/science.1150349

Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology letters*, *12*(12), 1394–1404.

Brooks, T. M., Mittermeier, R. A., da Fonseca, G. a. B., Gerlach, J., Hoffmann, M., Lamoreux, J. F., Mittermeier, C. G., et al. (2006). Global Biodiversity Conservation Priorities. *Science*, *313*(5783), 58–61. doi:10.1126/science.1127609

Buchanan, J. M., & Milton Z. Kafoglis. (1963). A Note on Public Goods Supply. *The American Economic Review*, *53*(3), 403–414.

Carpenter, S. R., Mooney, H. A., Agard, J., Capistrano, D., DeFries, R. S., Díaz, S., Dietz, T., et al. (2009). Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences*, *106*(5), 1305.

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C., & Daily, G. C. (2006). Conservation Planning for Ecosystem Services. *PLoS Biology*, *4*(11), e379. doi:10.1371/journal.pbio.0040379

Choe, C., & Fraser, I. (2008). A Note on Imperfect Monitoring of Agri-Environmental Policy. *Journal of Agricultural Economics*, *49*(2), 250–258. doi:10.1111/j.1477-9552.1998.tb01267.x

Choe, Chongwoo, & Fraser, I. (2008). Compliance Monitoring and Agri-Environmental Policy. *Journal of Agricultural Economics*, *50*(3), 468–487. doi:10.1111/j.1477-9552.1999.tb00894.x

Corbera, E. & Pascual, U. (2012) Ecosystem Services: Heed Social Goals. Science, 335, 655-656.

Cornes, R., & Sandler, T., (1996). The theory of externalities, public goods, and club goods. Cambridge, UK: Cambridge University Press.

DeFries, R. S., Houghton, R. A., Hansen, M. C., Field, C. B., Skole, D., & Townshend, J. (2002). Carbon Emissions from Tropical Deforestation and Regrowth Based on Satellite Observations for the 1980s and 1990s. *Proceedings of the National Academy of Sciences*, *99*(22), 14256–14261. doi:10.1073/pnas.182560099

Díaz, S., Hector, A., & Wardle, D. A. (2009). Biodiversity in forest carbon sequestration initiatives: not just a side benefit. *Current Opinion in Environmental Sustainability*, *1*(1), 55–60. doi:10.1016/j.cosust.2009.08.001

Dobbs, T. L., & Pretty, J. N. (2004). Agri-Environmental Stewardship Schemes and "Multifunctionality." *Applied Economic Perspectives and Policy*, *26*(2), 220–237. doi:10.1111/j.1467-9353.2004.00172.x

Ebeling, J., & Yasué, M. (2008). Generating carbon finance through avoided deforestation and its potential to create climatic, conservation and human development benefits. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1498), 1917–1924. doi:10.1098/rstb.2007.0029

Egoh, B., Reyers, B., Rouget, M., Richardson, D. M., Le Maitre, D. C., & van Jaarsveld, A. S. (2008). Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment*, *127*(1–2), 135–140. doi:10.1016/j.agee.2008.03.013

Feng, H., Kurkalova, L. A., Kling, C. L., & Gassman, P. W. (2006). Transfers and environmental co-benefits of carbon sequestration in agricultural soils: retiring agricultural land in the Upper Mississippi River Basin. *Climatic Change*, *80*(1-2), 91–107. doi:10.1007/s10584-006-9143-1

Fisher, B., Bradbury, R. B., Andrews, J. E., Ausden, M., Bentham-Green, S., White, S. M., & Gill, J. A. (2011). Impacts of species-led conservation on ecosystem services of wetlands: understanding co-benefits and tradeoffs. *Biodiversity and Conservation*, *20*(11), 2461–2481. doi:10.1007/s10531-011-9998-y

Frost, P. G. H., & Bond, I. (2008). The CAMPFIRE programme in Zimbabwe: Payments for wildlife services. *Ecological Economics*, *65*(4), 776–787. doi:10.1016/j.ecolecon.2007.09.018

Gibson, C. C., & Marks, S. A. (1995). Transforming rural hunters into conservationists: An assessment of community-based wildlife management programs in Africa. *World Development*, *23*(6), 941–957. doi:10.1016/0305-750X(95)00025-8

Goldman, R. L., Tallis, H., Kareiva, P., & Daily, G. C. (2008). Field evidence that ecosystem service projects support biodiversity and diversify options. *Proceedings of the National Academy of Sciences*, *105*(27), 9445 –9448. doi:10.1073/pnas.0800208105

Grieg-Gran, M., Porras, I., & Wunder, S. (2005). How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. *World Development*, *33*(9), 1511–1527.

Gustavsson, L., Karjalainen, T., Marland, G., Savolainen, I., Schlamadinger, B., & Apps, M. (2000). Project-based greenhouse-gas accounting: guiding principles with a focus on baselines and additionality. *Energy Policy*, *28*(13), 935–946. doi:10.1016/S0301-4215(00)00079-3

Hart, R., & Latacz-Lohmann, U. (2005). Combating Moral Hazard in Agri-Environmental Schemes: A Multiple-Agent Approach. *European Review of Agricultural Economics*, *32*(1), 75–91. doi:10.1093/erae/jbi002

Jackson, R. B., Jobbágy, E. G., Avissar, R., Roy, S. B., Barrett, D. J., Cook, C. W., Farley, K. A., et al. (2005). Trading Water for Carbon with Biological Carbon Sequestration. *Science*, *310*(5756), 1944 –1947. doi:10.1126/science.1119282

Kinzig, A. P., Perrings, C., Chapin, F. S., Polasky, S., Smith, V. K., Tilman, D., & Turner, B. L. (2011). Paying for Ecosystem Services—Promise and Peril. *Science*, *334*(6056), 603–604. doi:10.1126/science.1210297

Kinzig, A.P., Perrings, C., Chapin, F.S., Polasky, S., Smith, V.K., Tilman, D. & Turner, B.L. (2012) Response—Ecosystem Services: Free Lunch No More. Science, 335, 656-657.

Kleijn, D., Sutherland, & J, W. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, *40*(6), 947–969. doi:10.1111/j.1365-2664.2003.00868.x

Koch, E. W., Barbier, E. B., Silliman, B. R., Reed, D. J., Perillo, G. M., Hacker, S. D., Granek, E. F., et al. (2009). Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment*, *7*(1), 29–37. doi:10.1890/080126

Kohli, U. (1983). Non-Joint Technologies. *The Review of Economic Studies*, *50*(1), 209–219. doi:10.2307/2296966

Kohli, U. R. (1981). Nonjointness and Factor Intensity in U.S. Production. *International Economic Review*, *22*(1), 3–18.

Landell-Mills, & N., Porras, I., (2002). Silver Bullet or Fool's Gold? A Global Review of Markets for Forest Environmental Services and Their Impact on the Poor. London, UK: International Institute for Environment and Development (IIED). Retrieved June 26, 2012, from http://www.cbd.int/doc/external/iied/iied-silver-report-2002-en.pdf

Miles, L., & Kapos, V. (2008). Reducing Greenhouse Gas Emissions from Deforestation and Forest Degradation: Global Land-Use Implications. *Science*, *320*(5882), 1454–1455. doi:10.1126/science.1155358

Milgrom, P., & Roberts, J. (1990). Rationalizability, Learning, and Equilibrium in Games with Strategic Complementarities. *Econometrica*, Econometrica, *58*(6), 1255–77.

Millennium Ecosystem Assessment. (2005). Ecosystems and human well-being: current state and trends: findings of the condition and trends working group. Washington, DC: Island Press.

Miranda, M., Porras, I. T., & Moreno, M. L. (2003). The social impact of payments for environmental services in Costa Rica: a quantitative field survey and analysis of the Virilla Watershed. Markets for environmental services No.1. London, UK: International Institute for Environment and Development (IIED). Retrieved June 26, 2012, from http://pubs.iied.org/pdfs/9245IIED.pdf

Moxey, A., White, B., & Ozanne, A. (1999). Efficient Contract Design for Agri-Environment Policy. *Journal of Agricultural Economics*, *50*(2), 187–202. doi:10.1111/j.1477-9552.1999.tb00807.x

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R. E., Lehner, B., Malcolm, T. R., et al. (2008). Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences*, *105*(28), 9495.

Naidoo, Robin, & Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation. *PLoS Biology*, *4*(11), e360. doi:10.1371/journal.pbio.0040360

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., Chan, K. M., et al. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, *7*(1), 4–11. doi:10.1890/080023

Nelson, E., Polasky, S., Lewis, D. J., Plantinga, A. J., Lonsdorf, E., White, D., Bael, D., et al. (2008). Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proceedings of the National Academy of Sciences*, *105*(28), 9471 –9476. doi:10.1073/pnas.0706178105

Ozanne, A., Hogan, T., & Colman, D. (2001). Moral Hazard, Risk Aversion and Compliance Monitoring in Agri-environmental Policy. *European Review of Agricultural Economics*, *28*(3), 329–348. doi:10.1093/erae/28.3.329

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World Development*, *33*(2), 237–253.

Pattanayak, S. K., Wunder, S., & Ferraro, P. J. (2010). Show Me the Money: Do Payments Supply Environmental Services in Developing Countries? *Review of Environmental Economics and Policy*, *4*(2), 254–274. doi:10.1093/reep/req006

Pfaff, A., Kerr, S., Lipper, L., Cavatassi, R., Davis, B., Hendy, J., & Sanchez-Azofeifa, G. A. (2007). Will buying tropical forest carbon benefit the poor? Evidence from Costa Rica. *Land Use Policy*, *24*(3), 600–610.

Polasky, S., Nelson, E., Pennington, D., & Johnson, K. (2011). The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Environmental & Resource Economics*, *48*(2), 219–242.

Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1554), 2959 –2971. doi:10.1098/rstb.2010.0143

Raudsepp-Hearne, C., Peterson, G. D., & Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings* *of the National Academy of Sciences*, *107*(11), 5242–5247. doi:10.1073/pnas.0907284107

Robertson, N., Wunder, S., (2005). Fresh Tracks in the Forest: Assessing Incipient Payments for Environmental Services Initiatives in Bolivia. Bogor, Indonesia: Center for International Forestry Research (CIFOR). Retrieved June 26, 2012, from http://www.cifor.org/pes/publications/pdf_files/BRobertson0501.pdf

Sandler, T., (2004). Global Collective Action. Cambridge, UK: Cambridge University Press.

Shapiro, C., & Stiglitz, J. E. (1984). Equilibrium Unemployment as a Worker Discipline Device. *The American Economic Review*, *74*(3), 433–444.

Swallow, B. M., Sang, J. K., Nyabenge, M., Bundotich, D. K., Duraiappah, A. K., & Yatich, T. B. (2009). Tradeoffs, synergies and traps among ecosystem services in the Lake Victoria basin of East Africa. *Environmental Science & Policy*, *12*(4), 504–519. doi:10.1016/j.envsci.2008.11.003

Tallis, H., Kareiva, P., Marvier, M., & Chang, A. (2008). An Ecosystem Services Framework to Support Both Practical Conservation and Economic Development. *Proceedings of the National Academy of Sciences*, *105*(28), 9457–9464. doi:10.1073/pnas.0705797105

Taylor, R. (2009). Community based natural resource management in Zimbabwe: the experience of CAMPFIRE. *Biodiversity and Conservation*, *18*(10), 2563–2583. doi:10.1007/s10531-009-9612-8

Venter, O., Laurance, W. F., Iwamura, T., Wilson, K. A., Fuller, R. A., & Possingham, H. P. (2009). Harnessing Carbon Payments to Protect Biodiversity. *Science*, *326*(5958), 1368. doi:10.1126/science.1180289

Vincent, P. E. (1969). Reciprocal Externalities and Optimal Input and Output Levels. *The American Economic Review*, *59*(5), 976–984.

Wallace, K. J. (2007). Classification of ecosystem services: Problems and solutions. *Biological Conservation*, *139*(3-4), 235–246. doi:10.1016/j.biocon.2007.07.015

White B. (2002). Designing Voluntary Agri-environment Policy With Hidden Information and Hidden Action: a Note. *Journal of Agricultural Economics*, *53*(2), 353–360.

Wu, J., & Babcock, B. A. (1995). Optimal Design of a Voluntary Green Payment Program Under Asymmetric Information. *Journal of Agricultural and Resource Economics*, *20*(02). Retrieved from http://econpapers.repec.org/article/agsjlaare/30774.htm

Wu, J., & Babcock, B. A. (1996). Contract Design for the Purchase of Environmental Goods from Agriculture. *American Journal of Agricultural Economics*, *78*(4), 935–945. doi:10.2307/1243850

Wunder, S. (2007). The Efficiency of Payments for Environmental Services in Tropical Conservation. *Conservation Biology*, *21*(1), 48–58. doi:10.1111/j.1523-1739.2006.00559.x

Wunder, S., & Albán, M. (2008). Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics*, *65*(4), 685–698. doi:10.1016/j.ecolecon.2007.11.004

Zabel, A., & Roe, B. (2009). Optimal design of pro-conservation incentives. *Ecological Economics*, *69*(1), 126–134.